**The Economics of Species Conservation**

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**Abstract**: A quarter of all species around the globe are threatened with extinction; this paper reviews research in economics about how best to counter those threats. Normative research has developed useful tools for cost-effectively choosing areas of habitat to protect. Such work has also designed working-lands contracts that can induce efficient quantities and patterns of conservation on private lands. Positive research finds evidence that payments for ecosystem service programs are effective but legal protections for threatened species have a mixed record of success. Economists have also measured both the non-market benefits and the costs of species conservation. Emerging work is tackling the particular challenges to species conservation posed by climate change, demand for exploiting charismatic megafauna, and global population growth. Future research would do well to build on those three areas and to study distributional issues in the benefits and costs of species conservation.

**1. Introduction**

Many plant and animal species are at risk of extinction. The IUCN Red List indicates that 25% of all species are threatened with extinction (IUCN 2017). Figure 1 shows large numbers of threatened plants, fish, mammals, and birds by region in 2016, and patterns in the IUCN’s aggregate indices for species survival probabilities indicate that the threats are getting worse (Rodrigues et al. 2014).

These levels of endangerment are likely to be inefficiently high. Plant and animal species are valued by people for many reasons, including intrinsic and cultural value, recreation, and contributions to ecosystems that themselves produce ecosystem services like carbon sequestration and flood mitigation. Many of these values are public goods, and thus species conservation is underprovided by people making at least three types of private decisions about how to use the environment. People choose how much natural habitat to convert to agriculture, urban development, or other valuable human use. People make management decisions, such as invasive species control, that affect the quality of unconverted habitat for the species that depend on it. And people engage in hunting and collection of species in the wild.

Ideally, we would measure the external costs of actions that harm species and design policies to limit those actions to efficient levels and spatial patterns in the environment. However, implicit property rights often lie with land owners and users; this pattern is strong in the U.S., where the Endangered Species Act (ESA), which can force landowners not to damage habitat if a listed species relies on it, is controversial and under constant political siege. Thus, many efforts to protect species involve inducing voluntary conservation. Because species conservation is a public good, free-riding plagues fundraising for conservation through taxes and charities. Thus, economists are often limited to informing cost-effective conservation strategies – doing the best we can with the resources we have.

In this paper, we review research carried out by economists on several facets of species endangerment and conservation. First we discuss research that has studied optimal design of programs to counteract the market failures in species protection: habitat conservation programs, policies to encourage habitat maintenance and reduce incidental take, and management of hunting and collection. Second, we review work that has quantified the effects, benefits, and costs of some of these species conservation efforts. Third, we highlight the limited work that has been done on what are likely to be grand challenges to conservation in the future: climate change and the threats posed to species by a burgeoning human population. We end by identifying other key areas for future research.

**2. Economics of conservation policy design**

***2.1 Habitat conservation***

Economics can inform actions and policies that protect the habitat on which species depend for survival. The earliest and most prolific strain of such research has produced analyses of the best lands to set aside from productive use in nature reserves. What lands should be protected to accomplish the most good with a limited budget? Newbold & Siikamäki (2015) and Boyd et al. (2015) review research that aimed to provide tools to answer that question. We have learned that optimal reserve designs depend on spatial variation in many factors, such as the costs and conservation benefits of protecting parcels, and the threat of parcel conversion if conservation is not established. Fully optimal reserve networks can yield more benefits for the same costs than the results of simple heuristics because optimization routines take into account complementarity among sites (if one site already protects a species, the marginal value of other sites that do the same is lower).

Reserve-site selection (RSS) optimization algorithms are widely used by conservation practitioners. Users can optimize over single and multiple conservation objectives, allow total benefits to depend on the spatial configuration of protected parcels, and plan conservation actions that take place at multiple points in time. However, advances to this science can still be made. For example, RSS processes should account for behavioral spillovers. Bode et al. (2015) develop a stylized spatial simulation model of subsistence forest extraction where villagers both reduce total extraction and re-optimize their spatial pattern of extraction in response to creation of a protected area. The results show how the optimal network of protected areas must take spillovers into account, as protection of one site may displace environmental degradation into neighboring areas. These results are thought provoking but too stylized to be implementable; more work could be done in this vein. Additionally, few papers have used the tools of economists to incorporate uncertainty into RSS tools (Ando & Shah 2014). Instruments like portfolio theory can be used to reduce the uncertainty in the benefits gleaned from a network of protected areas (we discuss that work in section 4.1) but other strategies should be studied.

Choosing where to conserve habitat is just the first step toward species conservation. Programs must be designed to implement conservation plans, and such program design is complicated when priority lands for conservation are privately owned.

On rare occasions, political will permits habitat protection to be implemented with regulatory sticks rather than carrots. McConnell & Walls (2009) review tradable development right (TDR) programs that can be used in areas where the threat to species comes from development. The regulator sets a total desired level of development in an area, distributes development permits to landowners, and allows permits to be traded. Goals like species protection can be accomplished with TDRs by establishing protected “sending” areas in which owners can sell permits but not buy them, and “receiving” areas where they may buy but not sell. Such programs can accomplish desired conservation at lower total cost by allowing development to flow to places where the net private benefits of development are high. However, TDR programs can fail if transaction costs to trading are high and if the demand side of the market is thin because receiving areas are placed in areas already saturated with development.

Environmental Impact Fees (EIF) are the mirror image of TDR programs. Jiang & Swallow (2017) devise a program for use in ex-urban areas using EIFs on development to induce landowners to internalize the social value of damage to habitat if they develop their land. They use bioeconomic models of species conservation to show that EIFs can encourage landowners to internalize the environmental externalities of their development decisions in a setting where imperfect information and public funding constraints make it difficult to accomplish first-best conservation objectives with conservation payments alone.

If a species has been listed as threatened or endangered under the ESA, non-voluntary restrictions on land development and habitat use can be put in place. Such restrictions and direct government investment in species recovery are costly; Weitzman (1998) suggests a framework to determine priorities for species conservation. The benefits from investments in preservation of a species are given by the direct utility derived from that species, its contribution to diversity, and the enhanced survivability of the species that results from the investment (the difference in probability of survival with and without the investment). Benefits are combined with opportunity costs of investment into a formula that in principle could be used to rank species in order of priority for allocation of conservation resources.

However, the ESA is not designed to be efficient or cost-effective, provides no incentives for active stewardship of endangered species habitat, and can even give private landowners incentives to preemptively destroy habitat to evade regulation. Positive incentives and regulatory assurances can be more effective than just threats of regulation for inducing private landowners to help care for threatened species (Langpap 2006).

Indeed, where conservation is motivated by concern about an acutely endangered species that is not on the endangered species list but may be listed, voluntary conservation activities may be motivated by prelisting policies that encourage land owners and users to help prevent the listing and its rigid land-use restrictions from being necessary. Langpap & Wu (2017) develop a model in which landowners choose non-cooperatively whether to engage in preemptive habitat destruction or conservation. In their model, marginal increases in conservation reduce the probability of the species being listed if and only if conservation levels are above a threshold that varies with the conservation problem at hand. They show that while conservation payments may be needed in situations where landowners have strong incentives for preemptive habitat destruction, one may only need voluntary agreements and conservation credits to induce preemptive conservation in other cases.

When development taxes and regulations have inadequate political support and the species at hand are not endangered enough to invoke or threaten habitat use restrictions under the ESA, conservation success depends on our ability to convince private land owners to place their land in conservation status, either through fee-simple purchase or payment for conservation easements.

One challenge in voluntary conservation program design and administration is finding efficient levels of funds for conservation incentives. Several strategies have promise for minimizing free-riding in donations to conservation. For example, numerous papers have found that donations increase when provision-point mechanisms are used in which a donation threshold is established, often related to the total needed to accomplish a discrete public-good objective, and donors know they will be given their donation back if that threshold is not reached (Swallow 2013). Voluntary contributions can also be affected by the rule that governs the use of payments collected in excess of that needed to provide the good in question. Spencer et al. (2009) use lab experiments to test how different rebate rules perform at encouraging donations. They find that a proportional rebate rule (if total contributions exceed provision cost, a donor gets back a share of the excess donations in proportion to the size of their contribution relative to the total) induces revelation of true willingness to pay, both in total and by individuals.

The next challenge is to deploy conservation funds cost-effectively in private conservation incentives. Economic theory suggests that conservation incentives are subject to asymmetric information between landowners and conservation agencies. That may limit their cost-effectiveness through adverse selection if payments are accepted to protect lands that have low opportunity costs of conservation (e.g. less productive for agriculture) but, for the same reason, are at low risk of conversion. Hence, the agency may be paying for habitat that would not have been converted in the absence of the program, which therefore has low or no additionality. (Ferraro 2008). Many have studied how best to design conservation incentives in the face of asymmetric information and risk aversion (Segerson & Wu 2006; Smith & Shogren 2002; Shortle & Horan, 2001; Innes & Frisvold 2009). Given information asymmetries, a well-designed conservation incentive program would target payments on the basis of observable characteristics that are likely to predict the risk of habitat conversion, such as distance to roads and markets, assessed value, or soil type (Ferraro 2008). Furthermore, programs that use mechanisms such as auctions can induce self-reporting of private information and limit rent extraction by landowners.

Asymmetric information is not the only concern in cost-effective conservation payment design; uncertainty must also be taken into account. For example, Shah & Ando (2016) model conservation policy design when the returns from both land converted to commercial use and land in its natural state are uncertain. The latter uncertainty increases the option value to the landowner of delaying conversion, but multiple uncertainties also make landowners less willing to accept a permanent conservation contract and thus make temporary conservation contracts much less expensive.

Another strain in the literature on cost-effective voluntary conservation calls attention to problems of inducing landowner engagement with ecosystem-service provision when the benefits of an action on one parcel depend on adoption on other parcels, ecosystem-service production is characterized by minimum thresholds of adoption in a landscape, and social interactions affect landowner willingness to take part (Goldman et al. 2007; Hanley et al. 2012; Albers & Robinson 2015). Targeting strategies should take into account threshold effects, in which improvements in species status are achieved only after a minimum conservation investment has been reached, and correlation between effects on different species and even other environmental benefits. Otherwise, conservation expenditures may be excessively dispersed geographically, leading to lower impacts on habitat and species (Wu and Boggess 1999).

Lewis et al. (2011) compare the performance of several types of voluntary conservation incentives in the face of both spatially interdependent conservation benefits and asymmetric information, and find that very simple incentives can yield inefficient outcomes because outcomes are often very spatially fragmented. Lewis & Wu (2015) show that conventional markets for ecosystem services fail to produce efficient service provision. Efficient supply must induce landowners to internalize externalities among plots in ecosystem service production, and efficient market outcomes will usually require the government to act on behalf of service beneficiaries who may be spatially disconnected from service production.

Some research shows theoretically or provides experimental evidence that agglomeration bonuses (Banerjee et al. 2012; Drechsler et al. 2010; Parkhurst & Shogren 2007; Parkhurst et al. 2016) and other contract mechanisms (Dupraz et al. 2009; Reeson et al. 2011; Vogt et al. 2013; Warziniack et al. 2007) could be used to induce less fragmented patterns of conservation and thus produce conservation benefits more cost effectively. Polasky et al. (2014) develop a more complex conservation auction mechanism to implement the efficient quantity and spatial pattern of land conservation when spatial pattern matters and landowners have private information. In the classic Vickrey-Clarke-Groves auction mechanism each landowner has an incentive to bid exactly the opportunity cost of the contract to them because their payment does not depend on their bid. Polasky et al. (2014) harness the intuition of that mechanism with an auction in which landowners make conservation contract bids and the government offers payment to each landowner equal to the incremental benefits generated by their parcel given the other parcels that would be optimally selected with and without its inclusion in the network. This mechanism yields a first-best outcome in a simple numerical example and is intuitively promising. However, truth-telling may not be the dominant strategy if landowners expect the government to exploit revealed private cost information in future payment negotiations, and implementation may be challenged by transaction costs and social factors that limit landowners’ participation in auctions (Palm-Forster et al. 2016).

***2.2 Habitat maintenance and incidental harm***

Sometimes vulnerable species simply need their habitat to be protected from conversion. However, this passive approach to conservation will not correct market failures in situations where conservation benefits will not arise without action by land owners or users.

Contracts for conservation actions are hard to enforce because monitoring is difficult and enforcement is costly. Hence, participating landowners may not have the correct incentives to meet contract responsibilities (Ferraro 2008). Theory suggests that in the face of such moral hazard, payments should be output-based; that is, directly based on the environmental service provided, such as increase in population size or decrease in probability of extirpation for a specific species (Engel et al. 2008). Schlizzi & Latacz-Lohmann (2016) use principal-agent theory and lab experiments to study conservation practice payment design. They find that linking payment to outcome increases effort associated with conservation success but reduces landowner willingness to accept the contract in the first place. They also demonstrate how using a competitive tender process in selecting landowners with whom to contract can improve program performance, as landowners increase conservation effort in order to increase the likelihood that they are selected for the program.

One particular type of habitat maintenance is predator or competitor culling. A species can be endangered if the population of its predator or that of a competitor for its niche in the habitat grows too large. Melstrom & Horan (2014) use a numerical bioeconomic model of predator culling and habitat conservation in the case of endangered caribou in Quebec to explore the role culling can play in conservation of a prey species. They show that the optimal mixture of the two strategies depends on the impact of increased habitat on predator productivity; culling and habitat enhancement can be complements or substitutes in producing conservation benefits, and the optimal mix of strategies is likely to change with the temporal course of the species’ recovery.

Species can also be threatened if exotic invasive species (IS) invade their habitat, though Gurevitch and Padilla (2004) document that IS are a less common cause of endangerment than commonly thought. The spread of IS is spatial and stochastic, complicating the design of optimal management strategies; economists are engaged in ongoing research to understand how best to tackle IS control.

Sims & Finnoff (2013) use a dynamic programming model of decision making under uncertainty to model the optimal timing to start IS control; they find that a “wait and see” policy to controlling invasive species is only wise when the invader spreads slowly and when the nature of the species’ spread is highly uncertain. Spatial patterns of IS control are as important as timing. The spatial dimension of IS spread means that a land owner or manager rarely reaps all the benefits of the efforts they make. Theoretical/simulation models of spatial and dynamic IS spread and control processes in a fragmented privately owned landscape shed light on how to induce efficient quantities and patterns of IS control. With multilateral spatial externalities among producers who stand to lose from IS spread, cooperatives and Coasian compensation mechanisms may in theory accomplish nearly optimal levels of IS control, though transaction costs and information problems may hinder realization of that promise (Epanchin-Niell & Wilen, 2014; Liu & Sims 2016). In this setting, a Pigouvian tax on the presence of IS intended to provide landowners with incentives to engage in control can perversely suppress landowners’ incentives to control IS if the payment reduces the value of their land (Fenichel et al. 2014). The policy prescription for IS control changes when threats come from external sources like trade and tourism. Warziniack et al. (2013) use a general-equilibrium model to analyze how to mitigate the welfare loss from the externalities of recreation export that brings invasive species to a habitat. They find that taxes on visitors to induce efficient levels of visitation can be welfare improving. However, if the elasticity of demand for visitation is high and if tax revenue cannot be recycled, then local income and net welfare can be lowered by a tax aimed at controlling IS.

IS spread and control are not the only behaviors that affect species conservation success. Diverse papers have studied how to protect species by altering other behaviors of private agents; some illustrative examples are as follows. Endangered species can be harmed by the disruption and degradation that results from people visiting a habitat; Bednar-Friedl et al. (2012) develop a bioeconomic model of optimal tourism management that balances the value tourists gain from visitation (and fee revenue that can be used for management) with the resulting harm to a threatened species. Horan et al. (2011) review bioeconomic modeling research on protecting wildlife by stamping out disease; such efforts are complicated by interactions with privately-managed livestock. Ben Abdallah & Lasserre (2012) use dynamic programming to create an optimal logging rule that uses occasional brief logging bans rather than long-lived logging restrictions to protect the caribou in Canada from extinction with minimal lost timber value.

***2.3 Management of direct species extraction***

 Finally, hunting and collection directly harm species at risk of extinction. High profile examples include elephants, rhinoceros, tigers, and bears. Global willingness to pay to protect such species is large (Kontoleon & Swanson 2003) but difficult to mobilize. Numerous factors lie behind these extinction threats, but often the most prominent threat is unlawful hunting to avoid damage to agricultural output, to produce bushmeat for human consumption, and to supply illegal trade in live animals for collectors and body parts for ornamental and medicinal purposes.

The economics approach to studying hunting of vulnerable species usually relies on bioeconomic models that combine harvest functions and population dynamics, often combined with empirically-based numerical simulations, to predict how harvesting and stocks change in response to conservation policies. This type of analysis has shown that different policies can have qualitatively different effects on population dynamics, and hence that there is no single policy that is adequate for all species (Conrad & Lopes 2017).

 One type of policy seeks to help local communities to internalize more of the social benefits of protecting imperiled species through integrated conservation and development projects (ICDPs). These programs can generate income from conservation through resource harvesting, trophy hunting concessions, meat cropping, or shares of a park’s tourism profits. Fischer et al. (2011) examine Zimbabwe’s Communal Areas Management Program for Indigenous Resources (CAMPFIRE). They use a bioeconomic model in which the wildlife stock is affected by a public agency that determines hunting quotas, a local community that receives a share of benefits from wildlife tourism or from hunting licenses, and outside poachers. Wildlife is assumed to damage local agriculture but also generates rents for the community. Therefore, the community may choose to protect wildlife by deterring poachers. The authors conclude that benefit sharing does not necessarily generate both benefits and conservation incentives for the community. They show that whether this win-win outcome takes place depends on the types of profits derived from conservation and how much of them are shared. Johannesen & Skonhoft (2005) also use a bioeconomic model to examine a stylized ICDP with benefit sharing. They model strategic interaction between a park manager and the local community, which are both assumed to harvest wildlife. They find that a program in which the local community is given property rights over wildlife through a share of the park profit or of trophy hunting reduces poaching and increases conservation.

 A second category of policies takes steps to reduce demand for hunting and collecting. One element of demand for hunting is demand for food. Rentsch & Damon (2013) assess the potential effects of policies to reduce the quantity of bushmeat demanded in Tanzania by using price manipulation. They use data from protein consumption surveys to empirically estimate price, cross-price, and expenditure elasticities for various protein sources. Their results suggest that the most effective approach to reducing bushmeat consumption is to increase its price. They find that a 30% increase in bushmeat price would lead to a 34% reduction in consumption, and a 30% reduction in fish price is associated with a 20% drop in bushmeat consumption.

Local communities might demand wildlife hunting because of human-wildlife conflicts such as crop damage caused by wildlife near protected areas. When farmers hunt to reduce wildlife populations and thus the damages to their crops, governments and non-profit organizations frequently offer farmers compensation in the form of money, livestock, or seeds to reduce that motivation for hunting. Rondeau & Bulte (2007) use a dynamic model to examine the effects of such compensation on wildlife stock and on the welfare of local farmers. Their results suggest that a compensation policy aimed at reducing mortality from illegal hunting can actually reduce the stock of wildlife. Compensation increases the return to agriculture and thereby encourages agricultural expansion. The positive impact of reduced hunting on population stocks is offset by the negative impact on habitat, and the net effect can be negative.

Demand for rare species collection can be driven by consumers who want to possess animals because they are rare. Lyons & Natusch (2013) use data on wild harvests and sales of rare green pythons in the wild pet trade, finding evidence of the Anthropogenic Allee Effect (AAE) in which a species can be driven to extinction in a cycle fueled by strong preference for rarity. That cycle can be disrupted with calibrated regulation of harvest, and wildlife managers can avoid fueling the frenzy for species by not disclosing information about rarity.

 A third approach to hunting reduction focuses on supply-side policies. One option is to use supplies from captive-bred animals to meet demand for products derived from them, thereby lowering prices and making poaching unprofitable. A common example of this approach is tiger farming, with roughly 200 farms in China housing between 5,000 and 6,000 tigers (Denyer 2015). However, critics of farming argue that sales from captive animals legitimize consumption of these products, increasing demand and easing the laundering of illegal tiger parts. Economic analysis of wildlife farming suggests that the structure of the farming industry plays a critical role in determining whether this approach is beneficial to conservation. Damania & Bulte (2007) extend the basic poaching model to incorporate imperfect competition between poachers and farmers, and show that the effect of farming depends on the form of competition (Bertrand or Cournot). They conclude that farming could damage wild stocks if it takes place in a market characterized by aggressive price (Bertrand) competition. Likewise, Abbott & van Kooten (2011) use a bioeconomic model of wild tiger stocks, habitat, and trade to assess the potential of the captive tiger breeding industry to prevent extinction of wild tigers. They conclude that trade ban and anti-poaching enforcement approaches would have to be intensified to levels unattainable in practice to prevent tiger extirpation. However, allowing sales of products from tiger farms could be effective if sellers are granted a monopoly charter; a monopolist has an incentive to take actions against poachers to prevent them from dissipating monopoly rents.

 A related strategy is to use ex situ stocks of wildlife commodities, rather than products derived from farmed animals, to depress prices and discourage poaching. The same arguments raised against farming are relevant for selling stockpiles. There is concern that sanctioned sales of stockpiled African ivory would promote illegal killing of elephants. Mason et al. (2012) argue that holders of stockpiles may have an incentive to “bank on extinction” by contributing to the depletion of common stocks and accelerating the extinction process. The argument is that, as species decline, supplies from wild stocks may decrease and prices go up, making stockpiled products more valuable and putting additional pressure on remaining wild populations. They model the behavior of a speculator who has a large initial stock of a wildlife product, and show that it is optimal for them to deter the entry of poachers into the market either by depressing prices or reducing wild stocks. Which of these alternatives is chosen depends on initial wild and ex situ stocks. They apply their model to the case of the black rhino and find that, given the size of current stockpiles of rhino horns, the owner of these stocks would maximize profits by contributing to the collapse of wild populations.

**3. Empirical analyses of effects, benefits, and costs of conservation policy**

Research in economics has produced many guidelines for prioritizing conservation efforts and designing conservation policy. Economists have also found empirical evidence regarding the functioning of existing conservation programs. We begin this section with empirical evidence on how conservation policy decisions measure up to the prescriptions derived from economic theory, and the implications for the effectiveness of legislation designed to protect endangered species. Next, we review studies of how well programs such as protected areas and payments for ecosystem services work to preserve species habitat. We then review studies that attempt to measure use and non-use values of species. Finally, we discuss empirical measures of the costs and broader welfare impacts of conservation programs.

***3.1 Revealed priorities in the ESA and implications for its effectiveness***

Weitzman (1998) set forth recommendations for prioritizing conservation efforts among endangered species (discussed in section 2.1). How well does actual ESA administration reflect those recommendations? Metrick & Weitzman (1998) estimated the relationship between a proxy for the priority ranking of a species (government expenditures) and proxies for elements of the benefits and costs of conserving vertebrate species in the U.S. Benefits are captured by taxonomic category and body length (to capture the notion of charismatic megafauna), taxonomic uniqueness, and species’ endangerment levels (to capture survivability); whether a species’ recovery conflicts with economic activity is used as an indicator of opportunity cost. Their results suggest that charismatic megafauna effects (direct utility) are large, while survivability, diversity, and costs do not play the expected role in spending decisions. Specifically, more highly endangered species receive less spending, and species in conflict (higher cost) receive more spending. However, Dawson and Shogren (2001) find in a re-analysis of those data that, in addition to species’ non-scientific features or charisma, spending is driven by factors such as species’ long-term cultural value, the scientific knowledge base, value of the underlying habitat, and historical use of a species for game or commercial use.

Evidence thus suggests that species protection priorities may not account for the benefit and cost considerations suggested by economic theory. Has legislation designed to protect endangered species at least been effective? Detractors of the ESA note that only one percent of listed species have recovered sufficiently to be delisted, whereas supporters counter that the law has prevented the extinction of 99% of listed species (Langpap et al. 2018). Empirical evaluations of the effectiveness of various aspects of ESA implementation (listing, expenditures, Habitat Conservation Plans, recovery planning, and critical habitat designation) are reviewed in Langpap et al. (2018). Evidence indicates that listing is beneficial when supported by meaningful recovery expenditures, and that species-specific public investments decrease the probability that species decline and increase the probability that they remain stable and improve, even when they are not delisted due to recovery or extinction. Therefore, Langpap et al. (2018) argue that focusing the effectiveness debate on extinction and delisting may be misguided, and that assessments of the ESA’s efficacy must account for changes in species’ status between these two extremes. Otherwise, evidence of effectiveness may be overlooked.

 There are few empirical evaluations of the effectiveness of endangered species legislation in other countries. Adamowicz (2016) discusses the challenges of incorporating economic analysis into endangered species policy, and notes that under the 2003 Canadian Species at Risk Act (SARA), species remain at risk, more species are listed every year, and there are few delistings. Other assessments of SARA agree. Analyses of trends in species’ status find that improvement is rare and that most species evaluated either declined or remained stable. Specifically, the ratio of species that declined to species that recovered is roughly 2:1. Furthermore, the probability of improvement in status does not increase with time since listing under SARA (Favaro et al. 2014; Mooers et al. 2010).

In Australia, there is little evidence that the Environmental Protection and Biodiversity Conservation Act (EPBCA), passed in 1999, has been successful, with most listed species showing declining population trends (Taylor et al. 2011; Walsh et al. 2012). Botrill et al. (2011) evaluate the impact of recovery planning under EPBCA, and fail to find any short-term impact on species status. McDonald et al. (2015) review much of the literature on species conservation in Australia. They argue that overall more species are declining than improving, and conclude that conservation policy has failed to curb the loss of species.

Overall, there is evidence that some aspects of the U.S. ESA have been effective at promoting species recovery, but no convincing evidence of effectiveness for comparable laws in Canada or Australia. Comparative analysis of these laws and their implementation might be a fruitful avenue for future research to understand what underlies these differences in recovery outcomes.

***3.3 Habitat Conservation***

Next, we examine the effectiveness of two key strategies to conserve species by preserving their habitat: protected areas (PAs) and payments for ecosystem services (PES). PAs are one of the most commonly used tools for conservation of biodiversity and protection of threatened species. The number and size of protected areas has grown considerably (Watson et al. 2010), particularly in developing countries, where they cover roughly 15% of their combined land area (Miteva et al. 2012). PAs can reduce habitat destruction and fragmentation by restricting human use on large contiguous land areas. To be effective, PAs should be established on areas that represent important habitat for imperiled species and are threatened with conversion to other land uses (Miteva et al. 2012).

PES is an incentive-based conservation tool, in which buyers and sellers of ecosystem services (including biodiversity) enter into a voluntary agreement. Buyers agree to provide a payment, conditional on the supply of ecosystem services or on implementation of actions that generate them. There are hundreds of PES programs worldwide, particularly in developing countries (Arriagada et al. 2012).

Most studies of PAs and PES programs consider their impact on deforestation and forest degradation, which are good proxies for loss of species richness as forest conversion, degradation, and fragmentation negatively impact species (Lehtinen et al. 2003; Armsworth et al. 2004; Costa et al. 2005). Hence, policies that avoid or mitigate deforestation are expected to contribute to species recovery. PAs in Costa Rica, Thailand, Sumatra, Panama, and Indonesia have been found to reduce deforestation (Miteva et al. 2012). Sims (2014) finds that PAs have prevented forest loss and fragmentation in Thailand, where they are located in areas that are important to watershed protection, with high historical forest cover, protected scenic areas, and less high quality land. In contrast, Wendland et al. (2015) find that PAs in European Russia have had modest impacts. They note that this may reflect low forest disturbance rates in their study area, as well as most PAs being located far from threats.

Evidence on effectiveness of PES is less consistent. Alston et al. (2013) review studies from the perspective of wildlife habitat and biodiversity conservation. They note bird-focused PES programs in Cambodia that yielded more nests and birds, and Mexican programs to protect the Monarch butterfly that show evidence of additionality. In contrast, Pattanayak et al. (2010) review studies that find practically no impacts of PES programs on deforestation in Costa Rica, where payments were mostly not targeted and much of the land enrolled was at low risk of deforestation, and larger but still modest impacts in Mexico, where limited targeting based on expected deforestation took place. More recent assessments of Costa Rica’s PES program have reached almost opposite conclusions. Robalino & Pfaff (2013) evaluate the whole country and, in line with existing evidence, find small impacts on deforestation. In contrast, Arriagada et al. (2012) focus on a specific site with active targeting and cover eight years to allow for potential impacts to register. They find larger impacts in response to the PES program. This agrees with the theoretical insight that adequate targeting of payments may be an important driver of program outcomes.

Finally, a number of studies have examined the effectiveness of conservation programs in Mexico. Evidence from an early (2004) cohort of Mexico’s PES program (Alix-Garcia et al. 2012), as well as from later (2004-2009) cohorts (Alix-Garcia et al. 2015) suggests that total avoided deforestation is modest. However, given low deforestation among matched controls, the impacts are meaningful relative to what would have occurred in the absence of the program. Sims & Alix-Garcia (2017) evaluate PAs and PES in Mexico over the most recent decade and find that both programs have been effective in reducing predicted loss of forest cover.

To summarize, there is growing evidence that these conservation strategies can have a positive impact on species’ habitat, particularly in the context of forests in developing countries. The evidence for the effectiveness of PAs is more consistent, with positive impacts on forest cover ranging from roughly 15% to 25%, depending on the location. Recent evidence on PES also suggests meaningful positive impacts, with reductions in predicted loss of forest cover ranging from 11% to 50%. In general, the empirical evidence suggests that, consistent with theory, these conservation programs are more effective in terms of avoiding deforestation and thereby preserving species habitat if they are carefully targeted. PAs are more effective when placed in areas that are at higher risk of conversion. PES are more effective when payments are targeted on the basis of relevant observable characteristics correlated to a higher probability of deforestation.

***3.4 The Value of Species***

The studies discussed in the preceding sections assess the effectiveness of conservation programs in terms of their impact on species’ habitat or recovery status. Accounting for the value of the outcomes of conservation policies requires measuring the value society places on these species, and such valuation is difficult because of the non-market nature of these values. Economic studies have used a variety of methods to value threatened species or biodiversity, including revealed preference approaches like the travel cost method, or stated preference approaches like contingent valuation and choice experiments (Bartkowski et al. 2015). These studies have generated value estimates for over forty species, primarily mammals, fish, and birds (Wallmo 2015). Much of the literature on valuation of species is reviewed in a meta-analysis by Richardson & Loomis (2009). They conclude that the total economic value of species in the U.S. is affected by the type of species being valued, the suggested change in the species’ population, the charisma of the species, and payment frequency.

A number of recent studies have focused on the values attached to recovery of marine and anadromous species. Lew & Wallmo (2011) estimate annual household willingness to pay (WTP) for improvement in status for the Hawaiian monk seal and the small tooth sawfish, and Johnston et al. (2015) estimate annual household WTP for recovery and downlisting of the Puget Sound Chinook salmon and the Hawaiian monk seal. Boxall et al. (2012) estimate the economic benefits of marine protected areas and other management actions to improve the status of marine mammals in the St. Lawrence estuary of Quebec. They find that annual per household WTP for recovery of these species depends on the species affected, the size of the marine protected area, and the resulting restrictions on industry. Stefanski & Shimshack (2016) estimate annual household WTP for expanding marine protected areas in the Gulf of Mexico.

Two recent studies have focused on the value of bird species. Dissanayake & Ando (2014) estimate the values placed on grassland restoration and the marginal values of additional bird species in such areas. They find that households value diversity as such, but they place a higher value on actions that promote conservation of endangered species. Kolstoe & Cameron (2017) estimate the value of birding sites and the marginal value of additional species at these sites. They find that, for their baseline birding trip scenario, 95% of total WTP can be attributed to species richness.

In addition to estimating values for species, these studies also focus on innovations in methodology, choice of species or setting, or data. Lew & Wallmo (2011) test for sensitivity of values to the number of species being valued (scope), Johnston et al. (2015) develop tests for spatial heterogeneity in welfare estimates, and Lew & Wallmo (2017) evaluate the temporal stability of species values. Dissanayake & Ando (2014) are the first to estimate the value of grassland ecosystems, and they demonstrate how a choice experiment can guide ecosystem restoration projects. Kolstoe & Cameron (2017) illustrate the viability of using citizen data for valuation by using eBird data from the Cornell ornithology lab.[[1]](#footnote-1)

***3.5 Costs***

 The costs of conservation considered here include explicit costs such as land acquisition or restoration and management expenses, as well as opportunity costs such as the benefits forgone when land conversion or other economic activity is limited. We discuss the costs of legislation and other conservation approaches in Canada and the US, and then consider the impact of PAs and PES on welfare in developing countries.

Two studies have measured costs of conservation in Canada. Lawley & Towe (2014) examine the impact of habitat conservation easements on agricultural land values in the prairie pothole region of Manitoba. They find that the average eased parcel would sell for almost 13% less with an easement than without. Boskovic & Nostbakken (2017) use auction prices for oil leases in Alberta to estimate how these prices are affected by regulations to protect Caribou. They identify the effect of regulation on the value of natural resource development by comparing prices of leases on regulated land with unregulated but otherwise identical leases. They find that, on average, prices are lower by roughly 24% per hectare for leases inside caribou protection zones. They aggregate across leases and over years in their study period and estimate a total net present value cost of the regulation of at least $1.15 billion in foregone resource revenues for the government.

Costs related to implementation of the U.S. ESA are reviewed in Langpap et al. (2018). The Northwest Forest Plan imposed restrictions on the use of public forests, largely to conserve habitat for the Northern Spotted Owl and Marbled Murrelet; several papers estimate production possibility frontiers for species conservation and timber production and suggest that the marginal costs of conservation increase exponentially in that context. They review costs related to critical habitat designation and conclude that, while there is no evidence that critical habitat has affected land cover change, a few studies have identified impacts on housing markets in California.

Some of the costs related to the ESA are more indirect. For instance, recovery of carnivores such as the gray wolf has led to conflicts with livestock producers in the western U.S. Ramler et al. (2014) use data from ranches in Montana to measure the impact of wolf proximity and depredation on calf weight. They do not find evidence of an impact of wolf presence on weight, but they do find that calves on ranches where depredation occurs are lighter, which translates into losses in revenue from calf sales.

The welfare impacts of policy that sets aside land for conservation have been examined as well. The Conservation Reserve Program (CRP) removes environmentally sensitive land from crop production, thus lowering farm revenues. However, the program provides payments to participants which could positively affect household well-being (Chang et al. 2008). Empirical evidence suggests that CRP participation may indeed have mixed effects. CRP has been shown to increase average farmland values (Wu & Lin 2010). Additionally, CRP participation is associated with higher farm household savings at medium and upper quantiles of the income distribution, and higher consumption for lower percentiles of the distribution. However, CRP participation is associated with lower consumption for households in the medium and upper income percentiles, and with lower household income at all points of the distribution (Chang et al. 2008).

The welfare effects of PAs and PES in developing countries are also ambiguous *a priori*. While PAs can increase tourism revenue and induce infrastructure development, they can also constrain resource use, limit agricultural and extractive sources of revenue, reduce employment, increase uncertainty over property rights, and increase inequality (Ferraro & Hanauer 2011; Robalino & Villalobos-Fiat 2015; Sims 2010). Participation in PES programs is voluntary and provides households with financial transfers, and thus may raise income. However, restricted access to enrolled communal areas, as well as risk aversion or lack of financial literacy could have the opposite effect (Alix-Garcia et al. 2015; Arriagada et al. 2015).

There are few studies of the welfare impacts of PAs and PES, perhaps due to lack of reliable data on socioeconomic outcomes at the right scale and to non-random siting of PAs (Sims 2010). Nevertheless, the evidence suggests there are moderate positive impacts of these programs on surrounding communities and participating households. Sims (2010) studies PAs in Thailand and finds that an increase in the share of locality land protected raises monthly household consumption and lowers the poverty headcount ratio. The author argues that the most likely reason is that PAs generated sufficient local tourism revenue to offset income lost due to land use restrictions. Alix Garcia et al. (2015) identify small but positive poverty alleviation effects from the steady stream of income provided by Mexico’s Payments for Hydrological services program. Robalino & Villalobos-Fiat (2015) find that proximity to national parks increased wages of local workers in Costa Rica, mainly due to tourism-related effects. Ferraro & Hanauer (2011) confirm that Costa Rica’s protected areas had small positive impacts on poverty alleviation, even in high-poverty areas. An exception is the study by Arriagada et al. (2015), who find no effect of Costa Rica’s Payments for Environmental Services program on household wellbeing. This could be because participating landowners did not invest additional income in farm inputs or use it to enable a move off their farm.

Some studies emphasize the heterogeneity of effects and the potential for tradeoffs between conservation and socioeconomic outcomes. Robalino & Villalobos-Fiat (2015) consider spatial heterogeneity in the impacts of Costa Rica’s PAs; they find that, while there are positive wage impacts for workers living close to the entrance of a park, there are no impacts farther from the entrance. Ferraro & Hanauer (2011) consider heterogeneity in terms of demographic and biophysical characteristics. They identify a tradeoff, as characteristics associated with more avoided deforestation are also associated with less poverty alleviation. Alix Garcia et al. (2015) identify a similar tradeoff by finding larger socioeconomic impacts where deforestation risk is lower.

**4. Future Challenges**

Several critical challenges to species conservation are emerging. Here we describe some of the work done to define and address those challenges. However, in each case, much more work is needed. As humans become more numerous and technologically advanced, conservation research needs to consider solutions to: (a) the threats posed to species, and the complications posed to conservation planning, of climate change; and (b) species protection in urbanized areas.

***4.1 Climate change***

 An overwhelming plurality of climate scientists agree that while greenhouse gas reductions might curb the magnitude of climate change, greenhouse gas concentrations are so high that some warming of the earth is inevitable, and indeed has already begun and is affecting ecosystems (Hughes 2000). Biologists and ecologists have made clear that this warming poses a dire threat to the continued existence of many species (Thomas et al. 2004). Even species that survive will often be present only in very different places than they currently occupy (Parmesan & Yohe 2003). Scientists (including economists) are studying what kinds of actions and policies might mitigate the impacts of climate change on species survival, and how to make choices about those actions in the face of great uncertainty (Polasky et al. 2011).

 The seriousness of the climate threat to species survival is reflected in new willingness to contemplate the previously criticized practice of facilitated species migration. Many species that are threatened in their current locations might be able to thrive in new settings. However, natural species migrations in response to climate are often blocked by human barriers in the landscape: roads, buildings, and sea walls. Some species with limited mobility will not disperse as fast as the climate is changing, and thus may be extirpated before they can reach the new destination. Shirey & Lamberty (2010) assess whether assisted migration could legally be deployed as a tool to prevent extinction of species listed as endangered in the U.S. A multidisciplinary working group (Schwartz et al. 2012) review the ecological, ethical, and legal issues with assisted migration. However, economists have not applied their toolkit to study how this strategy could be utilized to protect species from climate change. Many of the analytical frameworks that have been brought to bear on reserve site selection and invasive species risk may be useful in this context.

Another area of research is a set of natural extensions to the optimal reserve site selection literature outlined in section 2.2. This literature developed tools that could use information about locations of species to choose optimal locations for conservation reserves to protect them. However, climate change uncertainty makes it very difficult to implement those site selection tools. How do you pick the best place to protect a species if you do not know where it will be in fifty years?

To answer that question, scientists have used decision theory (Moilanen et al. 2006) and complex applications of conventional site selection tools, like Marxan and Zonation, to data from multiple climate scenarios (Carvalho et al. 2011; Kujala et al. 2013). Ecologists have also called for increased emphasis on improving ecosystem connectivity and protecting climatic refugia (Groves et al. 2012). Economists have adapted portfolio choice tools from finance to this problem, choosing bundles of lands in a landscape (much like choosing bundles of stocks and bonds for retirement savings) to ensure that the overall conservation outcome of the set will not depend very much on what climate scenario comes to pass. Application of portfolio theory to a large set of diverse conservation problems indicates that portfolios can help reduce conservation outcome uncertainty in most cases (Ando et al. 2017). This tool is promising, and more work could be done to develop conservation portfolio tools that can accommodate spatial interdependence of the conservation benefits of protecting individual areas in a landscape.

However, if extreme climate events are likely, then the largest possible expected benefits of protected areas for climate-sensitive species are much lower, and portfolios of protected lands that are buffered against uncertainty have lower overall expected conservation success (Ando & Mallory 2012). Protected area portfolios are not a panacea. Indeed, there may be what Weitzman (2011) refers to as deep structural uncertainty about the probability distribution of future climate outcomes, with so much weight in the catastrophic tail of that distribution that the theoretical underpinnings of portfolio theory do not apply.

A complementary question is thus how to efficiently invest in management actions to mitigate the impact of climate change on species in their current locations. Ecologists have developed adaptation strategies to help ecosystems and their species to better endure the effects of climate change (Mawdsley et al. 2009). For example, the impact of climate change on wetland habitat in the Prairie Pothole Region can be reduced by changing farming practices on nearby lands (Voldseth et al. 2009). In another example, the rate of coastal wetland loss can be reduced by taking out bulkheads that prevent wetlands from retreating inland as sea level rises (Needelman et al. 2012). Such adaptation projects can increase the resilience of protected areas to climate change.

While ecologists are improving knowledge about how climate adaptation can be used to protect species from this threat, there is a gap in the economics research on this subject. The costs and benefit of adaptation strategies need to be estimated. The same tools of bioeconomic analysis used to study optimal adjustments in fishing, logging, and recreation behavior to protect species that are harmed by those activities could be used in studies of optimal implementation of climate adaptation strategies for species conservation.

Finally, economists and ecologists have already studied an indirect effect climate change might have on threatened species. International concern about the carbon emissions that spur climate change has created programs to incentivize massive forest conservation, such as REDD+. Such conservation could have benefits for forest species (Day et al. 2014; Harvey et al. 2010). However, those programs are targeted at carbon storage rather than species protection. Spatial priorities for those two objectives may not always align (Siikamäki & Newbold 2012) and afforestation may have serious negative consequences for water supply and biodiversity in some areas (Chisholm 2010). Biodiversity payments could be used in concert with REDD+ to ensure species protection (Busch 2013). However, legal institutions surrounding current climate policy frameworks pose challenges to integrating a focus on biodiversity into payments for forest conservation to store carbon (Phelps et al. 2012).

***4.2 Species conservation in a more populated world***

Species conservation is fundamentally a struggle to protect other species on earth from exploitation, habitat change, and other pressures created by people. All those pressures increase with human population, and that population is expected to grow to almost 9 billion people by the year 2050 (Cohen 2003). As people consume more water (Martinuzzi et al. 2014), use more land to grow food (Foley et al. 2011), and convert more natural habitat into subdivisions (Lawler et al. 2014), how do strategies for species conservation need to change?

Elements of the conservation movement has long prioritized pristine land and equated conservation with exclusion of human activity. However, that approach to conservation can be counter-productive, alienating local populations who could serve as powerful advocates for nature (Schwartzman et al. 2000). As humanity covers more of the planet, the conservation community necessarily pays more attention to conservation activity in urban areas and in landscapes where human coexists with nature (Kareiva & Marvier 2012).

Ecologists have carried out extensive research on how to accomplish conservation objectives in urban settings (Gehrt et al. 2010; Lepcyzk & Warren 2012; McCleery et al. 2014). Economists Sunding & Terhorst (2014) show that while species conservation in urban settings can be very costly, as it prevents high-value property development, the total cost of achieving species conservation objectives in urban areas could be reduced if local agencies permit more dense development near areas of restricted use. Conservation can be more cost-effective in urban areas if one alters other land-use policies in synchrony with setting some lands off-limits from development.

Conservation in urban areas can deploy strategies other than establishing dedicated protected areas. Urban gardens and agriculture can collectively represent significant acreages of habitat in urban areas (Rudd et al. 2002). They also serve to provide important connectivity between larger dedicated green spaces, though the value of gardens to species conservation is greater if policies and programs are established to encourage “wildlife friendly” management strategies (Goddard et al. 2010; Lin et al. 2015). Brownfields and other abandoned properties can provide important temporary habitat for urban biodiversity; even if individual parcels are redeveloped, churning of many parcels through developed and undeveloped status can provide a fluid set of areas that help to support many urban plants and animals (Kattawinkel et al. 2011).

Bauer et al. (2010a) study several different approaches to protecting amphibians in urban and ex-urban areas. Combining economic modeling with careful spatial meta-population models, they find that it is often important to regulate developed lands within the “matrix” of land within which protected areas around ponds are embedded in order to improve the capacity of the protected species to move between protected patches. Bauer et al. (2010b) find in the same setting that uniform wetland-protection policies are not cost-effective compared to spatially heterogeneous policies that account for variation in land value and ecological value. These studies of frog conservation in Rhode Island have some important general lessons. Fine scale spatial heterogeneity plays an important role in the economics of species conservation in urban area because the opportunity cost of conservation is large and highly variable. Also, much can be done in urban conservation just by making the area that is not completely protected from development less hostile to species in that urban ecosystem. Such efforts can have major ecological value and are less expensive than completely banning development on a parcel.

Indeed, various strategies can alter lands between protected areas to help with species conservation. For example, planting diverse vegetation and not mowing under power lines and along other utility rights of way can improve the size and connectedness of habitat for pollinators and birds (Russell et al. 2005). Even appropriately managed agricultural drainage ditches can provide important aquatic habitat if pollution flowing into them is not too great (Herzon & Helenius 2008).

**5. Conclusion**

Conservation of imperiled species remains a highly relevant policy issue worldwide, as wildlife continue to face enduringthreats of habitat loss, over-harvesting, and invasive species, threats which will only be compounded by climate change, a growing population, and urbanization. Economists have developed frameworks, laid out in a substantial body of work, to examine some of these issues in depth and suggest policy solutions. Other topics related to species conservation have not received as much attention by economists, and might benefit from application of the theoretical and empirical tools our discipline uses to model human behavior.

 Our review of normative studies suggests that the question of how to choose areas to protect as species habitat is a mature area of study. Nevertheless, innovation is still taking place in a variety of directions, as existing frameworks are being adapted to incorporate uncertainty and spatial issues. For instance, the adaptation of portfolio theory and real options theory are ongoing research topics. Research into how to modify behavior by using alternative conservation tools, such as tradable development rights, environmental impact fees, or prelisting conservation is ongoing as well, but would benefit from more attention.

 The positive analyses reviewed here indicate that there are relatively few studies of the effectiveness of species conservation laws. These studies have provided some evidence that such laws may have been effective in the U.S., but no evidence of their effectiveness in Canada or Australia. This suggests that there is much work to do, both in terms of assessing conservation legislation in other countries, as well as carrying out comparative analysis for the countries for which studies and data already exist. Other conservation strategies, such as PAs and PES programs, have been assessed as well, although in a limited number of locations. There appears to be consistent evidence of the effectiveness of PAs, and growing evidence of the effectiveness of PES, particularly for forest cover. Furthermore, most studies identify modest but positive impacts of these policies on the welfare of surrounding communities.

 We also reviewed two areas where species conservation will likely face growing challenges in the future: climate change and a growing population. While work is ongoing in both areas, for instance by adapting reserve selection tools to incorporate climate uncertainty, and developing conservation strategies for growing urban areas, these are topics that economists would do well to emphasize further. Finally, while we have developed a framework to evaluate benefits and costs of species conservation, economists could build on these structures to further examine the distribution of these costs and benefits. Carefully planned urban conservation can address some of the inequities of exposure to environmental degradation across racial and income groups (Jennings et al. 2012), and distributional issues are a key driver in the political economy of opposition to species conservation.

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**Figure 1.** Number of Threatened Species by Region in 2016

Source: World Bank. World Development Indicators Database.

1. This section focuses on estimates of use and non-use values of individual species. A further step is to combine these estimates with bioeconomic models using a production function approach to estimate broader values of species derived from their role in the ecosystem or link land use change with species productivity. Due to space constraints, discussion of this literature is outside the scope of this review. For other reviews, see Heal et al. (2005), Brander et al. (2006), Barbier et al. (2011), Barbier (2012). [↑](#footnote-ref-1)